LCA Methodology

Attributional and Consequential Environmental Assessment of the Shift to Lead-Free Solders

Tomas Ekvall[1]* and Anders S.G. Andræ[2]

[1] Division of Energy Technology, Chalmers University of Technology, SE-412 96 Göteborg, Sweden
[2] Swedish Microsystems Integration and Technology Centre (SMIT) and Department of Microtechnology and Nanoscience, Chalmers University of Technology, SE-412 96 Göteborg, Sweden

* **Corresponding author** (tomas.ekvall@me.chalmers.se)

DOI: http://dx.doi.org/10.1065/lca2005.05.208

Abstract

Goal, Scope, Background. As of July 1st, 2006, lead will be banned in most solder pastes used in the electronics industry. This has called for environmental evaluation of alternatives to tin-lead solders. Our life cycle assessment (LCA) has two aims: (i) to compare attributional and consequential LCA methodologies, and (ii) to compare a SnPb solder (62% tin, 36% lead, 2% silver) to a Pb-free solder (95.5% tin, 3.8% silver, 0.7% copper).

Methods. An attributional LCA model describes the environmental impact of the solder life cycle. Ideally, it should include average data on each unit process within the life cycle. The model does not include unit processes other than those of the life cycle investigated, but significant cut-offs within the life cycle can be avoided through the use of environmentally expanded input-output tables. A consequential LCA model includes unit processes that are significantly affected irrespective of whether they are within or outside the life cycle. Ideally, it should include marginal data on bulk production processes in the background system. Our consequential LCA model includes economic partial equilibrium models of the lead and scrap lead markets. However, both our LCA models are based on data from the literature or from individual production sites. The partial equilibrium models are based on assumptions. The life cycle impact assessment is restricted to global warming potential (GWP).

Results and Discussion. The attributional LCA demonstrates the obvious fact that the shift from SnPb to Pb-free solder means that lead is more or less eliminated from the solder life cycle. The attributional LCA results also indicate that the Pb-free option contributes 10% more to the GWP than SnPb. Despite the poor quality of the data, the consequential LCA demonstrates that, when lead use is eliminated from the solder life cycle, the effect is partly offset by increased lead use in batteries and other products. This shift can contribute to environmental improvement because lead emissions are likely to be greatly reduced, while batteries can contribute to reducing GWP, thereby offsetting part of the GWP increase in the solder life cycle.

Conclusions. The shift from SnPb to Pb-free solder is likely to result in reduced lead emissions and increased GWP. Attributional and consequential LCAs yield complementary knowledge on the consequences of this shift in solder pastes. At present, consequential LCA is hampered by the lack of readily available marginal data and the lack of input data to economic partial equilibrium models. However, when the input to a consequential LCA model is in the form of quantitative assumptions based on a semi-qualitative discussion, the model can still generate new knowledge.

Recommendations and Outlook. Experts on partial equilibrium models should be involved in consequential LCA modeling in order to improve the input data on price elasticity, marginal production, and marginal consumption.

Keywords: Attributional life cycle assessment; consequential life cycle assessment; lead-free; methodology for life cycle inventory analysis; solder paste

Introduction

Solder pastes are used by electronics manufacturers to provide electrical interconnections between electrical components and the Printed Wiring Board (PWB). Lead (Pb) has been used in solders for many years due to its low cost and high performance. Of the Pb present in European landfills, 40% originated from consumer electronics [1]. With the exception of solder alloys containing 85 mass % Pb, all other solders containing lead will be banned in the European Union (EU), due to the potential negative impact of Pb on the environment and on human health [2]. The inflow of Pb in alloys to EU landfills and waste incineration is to about 80% caused by solders used in electronics [3]. This calls for environmental assessments of Pb-free solders. These solder, although not new, have been little used to date since they have inferior mechanical and material properties for most electronics applications compared to conventional lead-tin (SnPb) solders. These properties include wettability, ductility, and higher corrosion resistance as well as higher reflow temperature.

Previous studies, by Griese [4], Deubzer [5–8], Hamano [9], Fukuda [10], Mihasecu [11] and Ascencio [12], on the one hand, and by Turbini [13], Graedel [14], Verhoef [15], Warburg [16], Geibig [17–18], Ku [19], and Ogunseitan [20], on the other, provide contradictory information concerning the environmental compatibility of Pb-free solders. The first group of authors are somewhat positive towards lead-free solders while the second is rather negative. However, all authors seem to agree on the higher reflow energy consumption of Pb-free, compared to SnPb, soldering. Some of the environmental assessments are traditional life cycle assessments (LCAs), while others focus on specific aspects of the solder life cycle.

In this paper, we distinguish between attributional and consequential LCA. Attributional methodology for life cycle inventory analysis (LCI) aims at describing environmentally relevant physical flows to and from a life cycle and its subsystems. Ideally, it should include average data on each unit process within the life cycle. The attributional LCI model does not include unit processes other than those of the life cycle investigated, but significant cut-offs within the life cycle can be avoided through the use of environmentally expanded input-output tables. Most previous LCAs resemble attributional LCA.

In contrast, consequential LCI methodology aims at describing how the environmentally relevant physical flows to and from the technosphere will change in response to possible changes made within the life cycle. A consequential LCI model includes unit processes that are significantly affected irrespective of whether they are within or outside the life cycle. Ideally, it should include marginal data on bulk production processes in the background system [21]. In a consequential LCI, allocation is usually avoided by means of system expansion [22]. A consequential LCI model can also include economic partial equilibrium models [23] and other tools designed to quantify specific causal relationships [24].

Several authors have made similar distinctions between the two types of LCA methodologies, although most authors employ different terms to denote them [21–22], [25–32]. The terms attributional/consequential were adopted in 2001 at a workshop on LCI electricity data in Cincinnati [33], but the term attributional had already been in use for several years.

1 Goal and Scope Definition

1.1 Goal definition

The main goals of this study were:

- to demonstrate and compare attributional and consequential LCA methodologies, and
- to contribute to the environmental assessment of a global shift from the common tin-lead wave solder paste (composition: 62% Sn, 36% Pb, 2% Ag) to one of the most common Pb-free reflow solder pastes (composition: 95.5% Sn, 3.8% Ag, 0.7% Cu).

1.2 Scope definition

Our environmental assessment includes an attributional and a consequential LCA. The function of solder paste is to keep electronic components attached to a PWB. The attachment requires a specific volume of solder paste. The functional unit chosen in both our LCAs is the volume of solder paste (approx. 300 mm^3) that is needed to mount components on to a specific, common PWB. This volume corresponds to 2.5 g SnPb solder. The corresponding weight of Pb-free paste is slightly lower because Pb-free solder has a lower density than SnPb (7.4 compared to 8.4 g/cm^3). The flux constituent and share of the weight is assumed to be the same for both solder pastes. The schematic product system structure of both solder paste pastes in the ALCI is shown in **Fig. 1**.

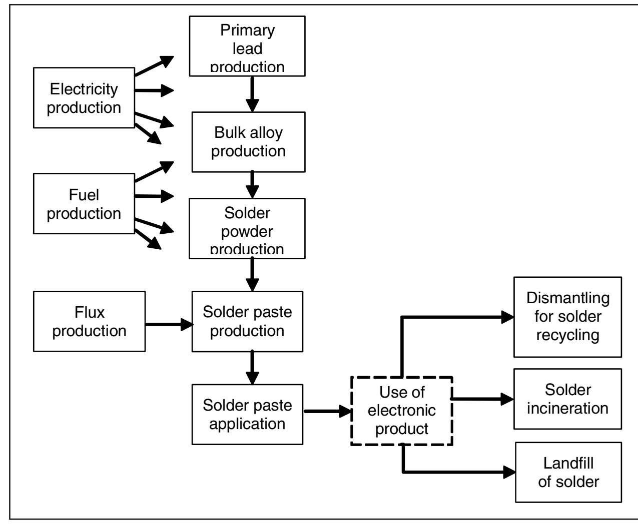

Fig. 1: Schematic structure of the SnPb and Pb-free solder paste attributional model

The use phase of the electronic product is excluded from the study, because the choice of solder has little impact on the energy demand of the electronic product. The electrical resistance of solders is insignificant compared to other resistance in the product (a typical resistance value for a solder joint is 0.01 Ohm and for a resistor 10,000 Ohm) [34]. The SnPb solder has a lower melting point than the Pb-free alternative (183ºC compared to 217ºC) but we assume the cooling of electronic products to be dependent on factors other than the solder melting point.

Dismantled PBAs have a positive economic value. For this reason, we do not regard it as waste but as a raw material inflow to the life cycles where the recycled metals are used. Thus, the recycling of metals from the solder belongs to subsequent life cycles and not to the investigated solder life cycle.

A description of the change in environmental burdens of the technosphere resulting from a shift from SnPb to Pb-free solder requires a CLCI that includes an analysis of how the relevant markets are affected. The preliminary structure of the CLCI system is presented in **Fig. 2**. The main difference compared to Fig. 1 is that the CLCI includes economic partial equilibrium models of the lead and scrap lead markets as well as the use of these goods outside the solder life cycle. The use of lead is the environmental issue addressed through the switch from SnPb to Pb-free solder. For this reason, the consequential study focuses on the lead flows.

For the partial equilibrium models we required information about the sensitivity of the supply and demand of lead and scrap lead to price fluctuations. In order to include the alternative use of lead outside the solder life cycle, we needed information on how sensitive different lead applications are to changes in the price of lead.

The solder shift increases electricity use, especially in the solder application. This means that less electricity will be available for other purposes. We tried to include a partial equilibrium model of the electricity and its alternative uses in the CLCI. There are several estimates of how price changes

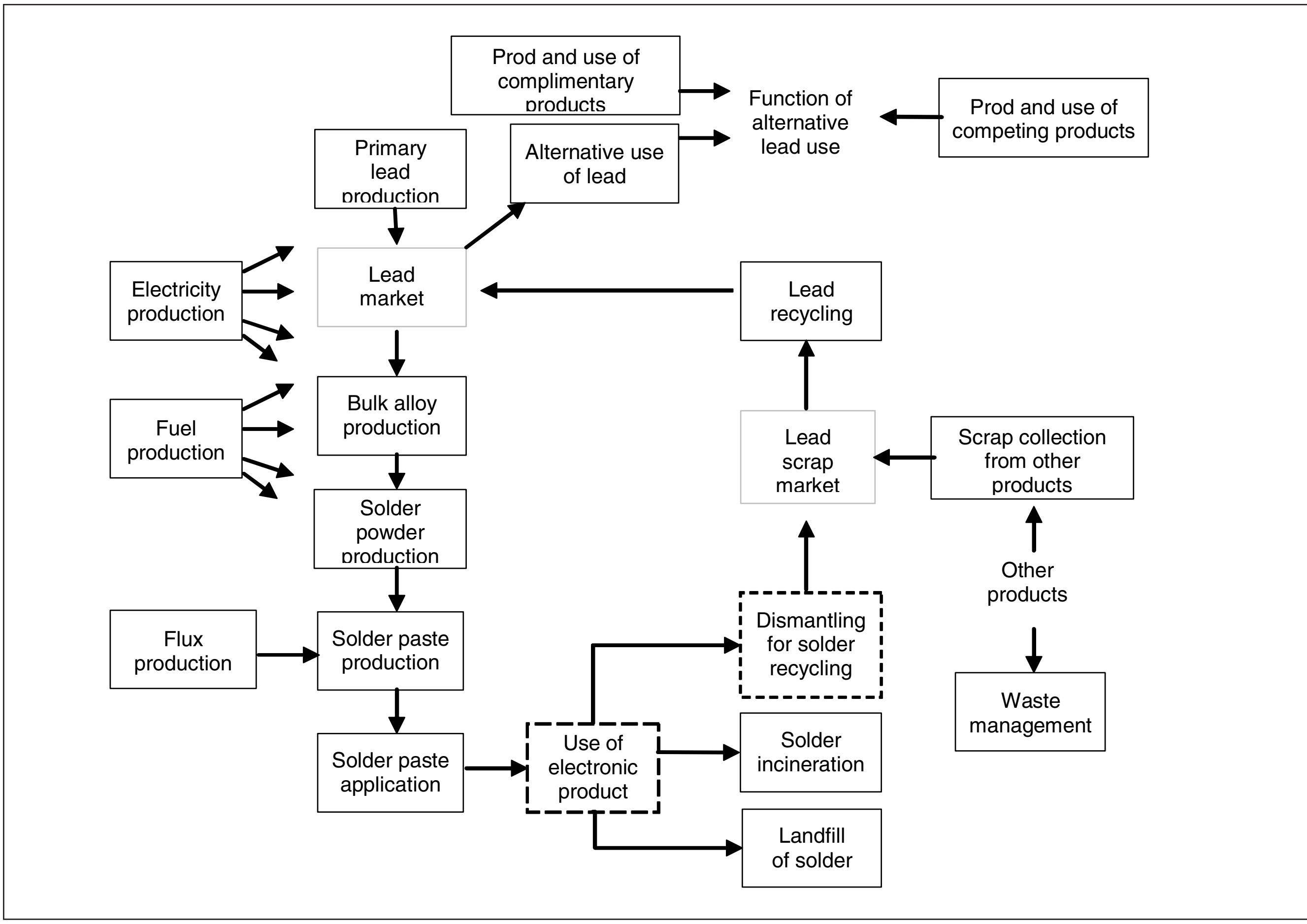

Fig. 2: Schematic structure of the consequential model of the switch from SnPb to Pb-free solder paste

affect electricity demand, but it is difficult to estimate how such changes affect electricity production on a global level. We also found no information on the marginal use of electricity, which means that we do not know which category of electricity use that will be reduced nor to what extent. For this reason, the electricity market was excluded from the CLCI.

An overview of initial data quality goals and data used is presented in **Table 1** (see next page). The study concerns the global shift from one solder paste to another. Solders are produced and applied worldwide, and metals, energy carriers and electronic products are traded on international markets. For this reason, the initial data quality goal was global averages for many ALCI subsystems. A change in demand for a metal, energy carrier etc. in the solder life cycle will affect the international markets at the margin. Thus, the initial data quality goal for many of the CLCI subsystems was global marginal data. The data actually used in the LCIs were more or less representative of data found in the literature or data from individual suppliers, due to the fact that global average and marginal data were not available (see Table 1, next page). A problem was that the available data sets included severe data gaps for lead emissions. In the end we used general estimates from the literature to calculate approximate lead emissions (see Sections 2.1 and 2.2.3).

2 Inventory

2.1 Attributional LCI

The start of the solder paste life-cycle involves the mining of silver, lead, tin and copper in different parts of the world. The metal bars are transported to a bulk alloy material production plant, where a wave solder of a specific composition is manufactured by melting the metals. Both primary and secondary metals can be used in bulk alloys [65], and about 23% of the tin, 50% of the lead, and 38% of the copper used worldwide are produced from secondary sources [66]. In the absence of metals recycling data, we used primary metals production data.

The bulk alloy is transported to powder production plants, where the solder is atomized into small, spherical balls. The next step is the solder paste production, where the powder is mixed with the flux system to form a paste, which is transported to electronics assembly plants. Here the first step is the screen printing, where the paste is applied to a Printed Wiring Board (PWB). This is followed by the reflow soldering process, where electronic components are placed on the PWB and the solder melted to form a Printed Board Assembly (PBA). Thereafter the PBAs are mounted inside the electronic products (EP), which are then shipped all over the world to be sold and used. After use, EPs including solder are transported to landfills, incineration or recycling

Table 1: An overview of initial data quality goals and data sources used in our study

	Attributional LCI		Consequential LCI	
Subsystem	**Initial data goals**	**Data used**	**Initial data goals**	**Data used**
Fuel production	Global average	Literature data [35–37]	Global marginal	Literature data [35–37]
Electricity production	Global average	Global average mix of technologies [38] Environmental data from the literature [35–37], [39–44]	Global marginal	Global average mix of technologies [38] Environmental data from the literature [35–37], [39–44]
Metals production	Global average for specific metals	Literature data on primary metals production [45–48]	Global marginal for specific metals	Literature data on primary metals production [45–48]
Lead market	n.a.	n.a.	Long- term global own price elasticity of demand and supply	Semi-quantitative estimates
Alternative use of lead	n.a.	n.a.	Global marginal gate-to-grave data on affected product mix	Gate-to-gate data on lead-acid battery assembly [49]
Production and use of complementary products	n.a.	n.a.	Global marginal cradle-to-grave data on actually complementary products	Cradle-to-gate data on photovoltaic cells [50]
Production and use of competing products	n.a.	n.a.	Global marginal cradle-to-grave data on actually competing products	Emissions from electricity production in diesel generators [37], [51–52]
Bulk alloy production	Global average for solder alloy	Data from one supplier of a different alloy [53]	Global average for specific solders	Data from one supplier of a different alloy [53]
Flux production	Global average for solder flux	Data from environmentally extended input/output table [54–55]	Global average for solder flux	Data from environmentally extended input/output table [54–55]
Solder powder production	Global average for solder powder	Data from one supplier of a different metal powder [56]	Global average for specific solders	Data from one supplier of a different metal powder [56]
Solder paste production and application	Global average for specific solders	Literature data [57–61]	Global average for specific solders	Literature data [57–61]
Dismantling for solder recycling	Global average for electronic products	Primarily from one electronics dismantler [62]	Global average for electronic products	Primarily from one electronics dismantler [62]
Solder incineration	Global average for specific solder metals	Data from one waste incinerator [63]	Global average for specific solder metals	Data from one waste incinerator [63]
Landfilling of solder	Global average for specific solder metals	Data from one landfill [64]	Global average for specific solder metals	Data from one landfill [64]
Scrap lead market	n.a.	n.a.	Long-term global own price elasticity of demand and supply	Assumption (see Section 2.2.7)
Waste management of other products	n.a.	n.a.	Global mix of regional marginals	Data from one landfill [64]
Lead recycling	n.a.	n.a.	Global marginal	Data from one site [53]

n.a.: not applicable

facilities. We estimated that, globally, 10% of old EPs are recycled, 14% incinerated, while 76% end up in landfills [17].

Data on Pb emissions were absent in the available datasets on most activities in the life cycles. Instead, we calculated the Pb emissions from estimates in the following literature. Brady & McManus [67] stated that 32 ktonnes of lead was used in solders and other alloys in the 15 EU Member States in 1998. They also stated that 30 tonnes of lead was emitted from alloys in the same countries in 2000. These data indicate that approximately 0.1% of the lead used for solders is emitted into the environment. Tukker et al. [3] estimated that 2/3 of the Pb emissions from alloys are emitted to air and the rest to soil.

Table 2 presents selected results from the ALCI. The electricity consumed in the life-cycles is 1.52 MJ for SnPb and 1.70 MJ for Pb-free solder. The Pb calculations in the ALCI are based on the sources above, which means that Pb emissions from the production of, e.g., electricity and ancillary materials are not included in these results.

Table 2: Selection of results from the ALCI of SnPb and Pb-free solder, and from the CLCI of the SnPb solder. Our CLCI of Pb-free solder is identical to the ALCI of the same product due to lack of marginal data (see Section 2.2.8). Functional unit: 300 mm³ of solder (cf. Section 1.2)

Substance	Category	Unit	SnPb ALCI	Pb-free ALCI	ALCI: Pb-free – SnPb	SnPb CLCI	CLCI: Pb-free – SnPb
Coal	Resource	g	83	93	10	82	10
Oil	Resource	g	12	14	2	15	-0.3
Natural gas	Resource	g	12	14	1	12	1
CH_4	Emission to air	g	0.78	0.87	0.09	0.78	0.09
CO_2	Emission to air	g	228	254	26	234	20
NO_x	Emission to air	g	0.9	1.1	0.2	1.1	0.05
SO_x	Emission to air	g	1.8	2.1	0.4	1.7	0.4
Pb	Emission to air	g	1e-3	0	–1e-3	9e-4	-9e-4
Pb	Emission to soil	g	5e-4	0	–5e-4	5e-4	-5e-4

2.2 Consequential LCI

In the CLCI we estimated the changes in production (ΔS) and alternative use (ΔD) of lead and scrap lead based on the estimated own price elasticity of supply (η_S) and demand (η_D) for each of the goods:

$$\eta = \Delta Q/Q * P/\Delta P \qquad (1)$$

where ΔQ is the change in the quantity produced or demanded of a specific good, caused by a change in its price (ΔP). Hence,

$$\Delta S/\Delta D = \eta_S/\eta_D \qquad (2)$$

In most cases, both supply and demand are more elastic in the long term than short term, the reason being that more factors can adapt to price changes in the long term: the production capacity of mines and power plants, the stock of household appliances, etc. We focus on long-term elasticity, as we believe that environmental systems analyses are primarily conducted due to concern about the long-term future environment. In order to describe how the environmental burdens of the technosphere are affected by a shift from SnPb to Pb-free solder, we tried to identify the marginal production technologies for lead. Such technologies can be identified using, for example, a five-step procedure presented by Weidema et al. [68]. In addition, we wanted to identify the marginal application of lead outside the solder life cycles. It is unclear whether a procedure exists for identifying the marginal use of a specific good.

2.2.1 Marginal electricity production

The ideal electricity data for the consequential LCI are a weighted average of long-term marginal electricity data over all electricity markets affected by the change in solder composition. This is difficult to identify, partly because decisions on different technologies are on the agenda in different countries. At a European level, a phase-out of nuclear power is under discussion in Sweden and Germany, while the construction of a new nuclear power plant is being considered in Finland. Investments in several natural gas power plants have recently been debated in, e.g., Italy, while wind power is expanding in, e.g., Germany. At a global level, marginal electricity is sometimes assumed to be based on coal, because coal is abundant and coal power is thus cost-efficient. However, climate-change policy may set a global 'ceiling' on this technology or at least make it too expensive for new investments in many parts of the world. Due to lack of data on the weighted average of marginal electricity, we used data on global average electricity production to model electricity production in the consequential LCI (see Table 1).

2.2.2 Marginal metal production

In the CLCI we attempted to identify marginal lead production. Globally, primary lead production can be divided into four main processes; ore mining, ore concentration, smelting, and refining [69]. Mining has inherent constraints depending on the location of lead ore. The other three stages are unlikely to be constrained and the choice of technology is related to cost. The new direct smelting processes seem to be more efficient than the more common blast-furnace methods [70], but a combination of several technologies is currently used, and it has not been possible to identify a clear-cut marginal technology for each of the four stages. In the absence of data on marginal lead production, we used the same input data as in the ALCI. This also applies to the other metals (see Table 1).

2.2.3 Marginal lead use

Most lead (74% worldwide in 1999) is used in lead-acid batteries. Leaded glass is used, for example, in the screens of television sets and computers to protect the viewer from the X-rays generated by the cathode ray tube. Lead sheet is used for complete roofs, the vertical cladding of walls, and construction details. Nowadays, lead piping is primarily used in the chemical industry. Ammunition accounts for 1–3% of global lead consumption.

Batteries are used in motor vehicles and as a backup power supply in the event of power failure. Lead-acid batteries can compete with NiCd batteries and new electricity storage systems such as metal hydride batteries. These are currently more expensive than lead-acid batteries [71], but may be a feasible alternative for many applications in the future. Lead-acid batteries can also be used in emerging markets [71]:

– for electric powered vehicles, which could compete with diesel and gasoline cars in the future,
– as a power supply (in combination with photovoltaic cells etc.) in remote areas, competing with diesel generators etc., and
– as peak-load power supply (in combination with base-load power stations), competing with oil and gas turbines, etc.

Lead accounts for at least 30 percent of the total production cost of lead-acid batteries [72]. One of the biggest challenges confronting stationary lead-acid battery producers is the fluctuating price of lead [73]. If the price of lead drops, lead-acid batteries can be sold at a lower price. This may affect the use of such batteries in applications where they compete with nickel-cadmium batteries and in all emerging markets. A reduced lead price may also affect the development of new alternatives such as metal hydride batteries.

A change in the price of lead can also affect the use of this metal in certain other products, particularly in lead sheet for roofing or wall cladding. A large quantity of lead is required for this purpose and the metal does not provide apparent functional advantages. In other lead products, the price of lead appears to be less important. In some cases the cost of lead is a small part of the total production cost while, in others, lead has important functional advantages over other materials, making the demand for lead less sensitive to changes in the lead price.

Our conclusion is that a switch from leaded to unleaded solders is likely to result in an increased use of lead in a mix of other products. This mix is probably dominated by batteries, because most of the lead is currently used in batter-

ies, and a change in lead price may significantly affect the current and future competitiveness of batteries. In the foreseeable future, the lead price is most likely to affect the use of lead-acid batteries in applications where there is stiff competition with other alternatives.

In our LCI model, the alternative use of lead is acid-lead batteries for use as a power supply in combination with photovoltaic (PV) cells. The lead-acid batteries and PV cells generate electricity in remote areas and we assumed that they replace diesel generators. The assumption that the marginal use of acid-lead batteries are in PV systems that replace diesel generators is an optimistic scenario for the shift to Pb-free solder, from the perspective of climate change. It is optimistic because, in this scenario, the shift to Pb-free solder results in an increased use of PV systems that replace fossil fuel. Such a scenario is relevant because the ALCA indicated that the solder shift increases the GWP of solder life cycles. Using an optimistic scenario in the CLCA makes it possible to test the robustness of this result. Referring to Fig. 2, this means that the alternative use of lead is assumed to be acid-lead batteries, the complementary product to be PV cells, and the competing product electricity produced by diesel generators. The production, use and waste management of these products are included in the CLCI.

As in the ALCI, the Pb emissions from the alternative use can be calculated from estimates in the literature. According to Brady & McManus [67], 919 ktonnes of lead was used in batteries in the 15 EU Member States in 1998, and 260 tonnes of lead was emitted in 2000. This indicates that 0.03% of the lead used for batteries is emitted to the environment. Tukker et al. [3] estimated that 92% of the Pb emissions from batteries are emitted to air, and the rest to soil.

2.2.4 Lead market

Since quantitative estimates on η_S and η_D are difficult to find for lead, we made our own estimates based on the following, qualitative discussion. In the short term, the elasticity of demand (η_D) is low, due to the fact it is hard for a manufacturer to substitute lead for another material [74]. In the long term, the elasticity of demand is higher, since the marginal lead products (batteries, roofing and wall cladding) has the potential to replace competing products. Nevertheless, we expect the elasticity of demand to be fairly low in the long term as well, because a change in lead price will only affect a part of the production cost of lead products.

The elasticity of supply for metals, η_S, is low in the short term because mines and mills are designed for a certain throughput [74]. In the long term, the elasticity of lead supply can also be assumed to be fairly low, because lead is typically a co-product of zinc and other metals. This means that a change in the price of lead will affect only part of the revenues of the mines and mills.

When the elasticity of both supply and demand are low for a specific good, we would expect its price to fluctuate greatly: when supply and demand are not in equilibrium, large price changes are required to restore the balance. Such a fluctuation can be observed at, e.g. the London Metal Exchange (LME); in 1996 and again in June 2004 the lead price exceeded 800 USD/tonne but dropped to 400–500 USD/tonne from the autumn of 1999 until autumn 2003 [75]. It is possible that these fluctuations reflect low short-term price elasticities. However, large quantities of lead are bought not for production but for investment purposes. Such investments may have a greater impact on short-term changes in metal prices than the actual production and consumption of metals [76].

The above arguments indicate that η_S and η_D are fairly low in the long term. The LME data indicate that this may be the case at least in the short term. We see no reason to claim that either one is higher than the other. In our LCI model, the supply and demand for lead are equally elastic: $\Delta S = -\Delta D$. This means that for each tonne of lead that is eliminated from the solders, the production of lead is reduced by 0.5 tonne and the use of lead in other products is increased by 0.5 tonne. The uncertainty in this part of the model is large.

2.2.5 Marginal lead recycling

Scrap lead can be transported long distances to the recycling process [77]. Secondary lead smelting and refining production technologies are likely to be unconstrained and the choice will be based on cost. It has not been possible, within this study, to identify a specific marginal lead recycling technology. Instead, we used data from one production site (see Table 1).

2.2.6 Marginal lead waste management

Globally, about 75% of all products containing lead (mainly batteries) are recycled [78]. Lead products that are not recycled are deposited at landfills, either directly or after passing through an incineration plant. The marginal lead waste management for lead products other than electronic products, is likely to be landfill, as it is cheaper than incineration. The Pb entering the scrap lead market originates from printed wiring boards where SnPb solder has been used and from other recycled lead products. Moreover, there is an important difference between simple dumps and properly managed landfills with leachate control. Ash containing lead from incineration plants should also be taken into account. The toxic ash must be managed in an appropriate manner [79].

2.2.7 The scrap lead market

We assumed that the collection of scrap PBAs with SnPb solder affects not only the use of scrap lead but also the collection of scrap lead from other products in accordance with eq. 2. Quantitative estimates on η_S and η_D are difficult to obtain for scrap lead. We have assumed that they are equal, i.e. that the demand and supply side of the market are equally affected by small changes in the price of scrap lead. This means that 50% of the lead from scrap PBAs replaces scrap lead from other products and that the remaining 50% contributes to an increase in total lead recycling.

2.2.8 Consequential LCI results

The CLCI of SnPb solder differs from the ALCI in that it includes the lead market, the alternative use of lead, and its complementary and competing products (see Figs. 1 and 2). The CLCI also includes lead recycling, the scrap lead market, and waste management of competing scrap lead sources. The CLCI of Pb-free solder is identical to the ALCI of Pb-free solder. The CLCI should ideally include marginal data for many unit processes while the ALCI should include average data. However, due to the lack of marginal data, we used the same input data in both Pb-free models (see Table 1). Table 2 presents selected results from the CLCI. The Pb results reflect emissions from lead products only, which means that Pb emissions from the production of, e.g., electricity and ancillary materials are not included in these results. The Pb results in the CLCI of SnPb solder are dominated by emissions from the solder life cycle (1.1 mg/f.u to air and 0.5 mg/f.u. to soil). The avoided emissions from the battery life cycle are much smaller: 0.2 mg/f.u. to air and 0.02 mg/f.u. to soil).

3 Impact Assessment

The impact category selected in this study is GWP_{100}, as this category has been found to be robust and reliable in previous LCA studies. The characterisation factors are presented in **Table 3**, and the total ALCA results are presented in **Fig. 3**. **Fig. 4** shows the contribution from different life-cycle stages of the Pb-free solder. Solder paste application dominates the results, as it requires large quantities of electricity. **Fig. 5** presents the contribution from different parts of the consequential model on SnPb solder. Again, the solder application dominates the results.

Fig. 6 presents an estimate of how a change from SnPb to Pb-free solder affects the greenhouse gas emissions from the technosphere. According to the results of the model, the emissions increase because of the increased production of tin in the Pb-free system and because of the greater amount of energy required for application of the Pb-free solder. In our study, the shift is assumed to result in more lead being used to produce batteries which, when combined with photovoltaic cells, will replace fossil fuel in electricity generation. This can be regarded as an optimistic scenario, but the reduction in emissions from diesel electricity is still not sufficient to outweigh the increase in emissions caused by solder applications and tin production. The reason for this is that little electricity is produced by the photovoltaic-plus-battery system: only 12 kWh for each kg of lead used.

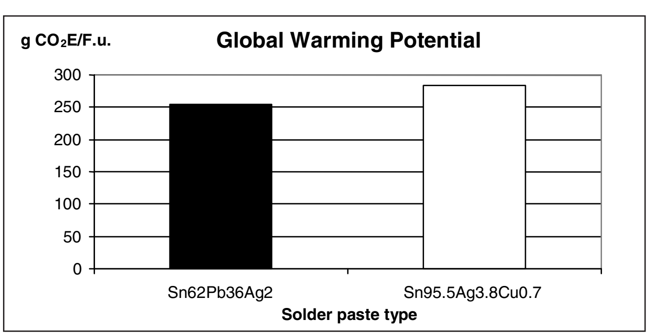

Fig. 3: Total GWP_{100} results of the ALCA

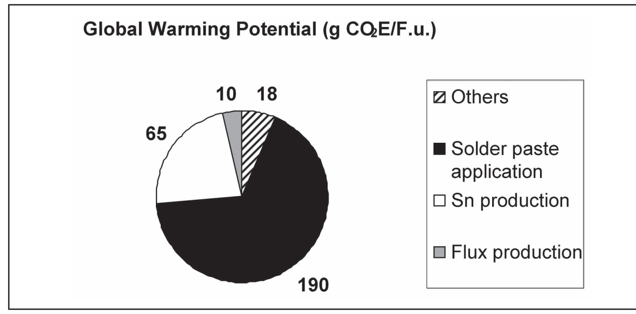

Fig. 4: Contribution to GWP_{100} from different life cycle phases in the ALCA of Pb-free solder

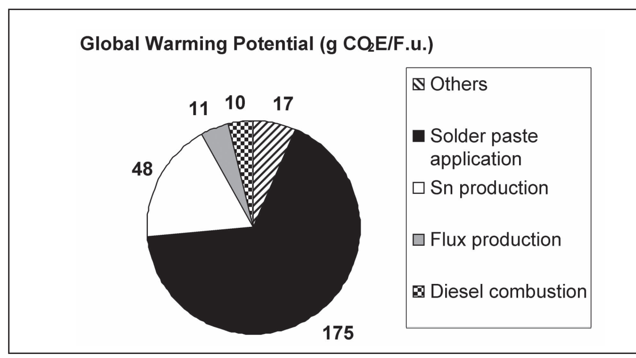

Fig. 5: Contribution to GWP_{100} from different life cycle phases in the CLCA of SnPb solder

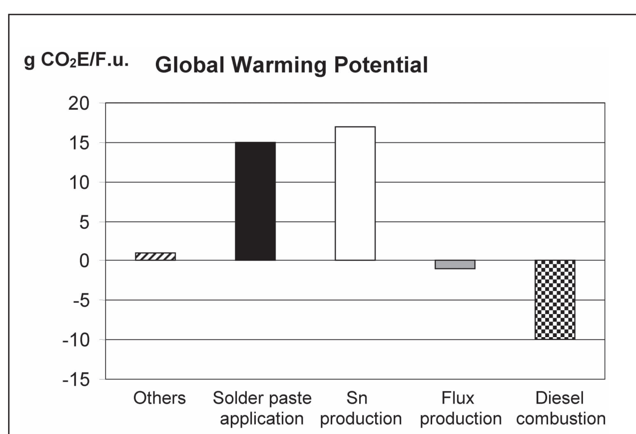

Fig. 6: The consequential GWP_{100} results obtained when subtracting the CLCA results for SnPb solder from the CLCA (=ALCA) results for Pb-free solder

Table 3: Characterization factors for GWP_{100} · $CO_2E = CO_2$-equivalents

Substance	Equivalence factor ($g\ CO_2E/g$)	Reference	Approximate share of GWP total in the SnPb and Pb-free life cycles (%)
CO_2	1	[80]	90
CH_4	23	[80]	7
NO_x	7	[81]	2.5

4 Interpretation

Based on global warming considerations, the energy required for solder paste applications was the most significant part of the solder paste system, regardless of whether SnPb or Pb-free solder was investigated and regardless of whether ALCA or CLCA methodology was used. The results indicate that the most important differences between SnPb and Pb-free solders are the tin production, the solder application, and the diesel replaced by the alternative use of lead. The last aspect can only be identified by means of a CLCA.

Overall, the data quality in the study is poor. The uncertainty is very large in the area of the environmental consequences of alternative lead use, which, according to the CLCA results, is one of the important issues. The data quality is especially poor in the model of the scrap lead market and waste management of competing sources of scrap lead. However, this part of the CLCA model for SnPb solder does not have a major bearing on the results.

Our results indicate that silver production is not an important part of the solder systems. It may, in fact, be more important in reality. In the literature used, silver production was modelled as a by-product of copper production [46]. In reality, 31% of silver is produced from zinc sources, 28% from virgin silver sources and 25% from copper sources [82].

The attributional LCA reveals the obvious fact that the shift from SnPb to Pb-free solder means that lead is essentially eliminated from the solder life cycle. A consequential LCA can show to what extent the shift leads to an overall reduction in the use of lead, how lead use will increase in other life cycles and, hopefully, in what products the use of lead will increase when the metal is eliminated from solder paste. Our results indicate that a shift of lead use from solders to other products reduces the Pb emissions and the associated risk to human health and ecosystems.

5 Conclusions and Discussion

The shift from SnPb to Pb-free solder is likely to result in reduced lead emissions and increased GWP. The alternative use of lead will partly offset the reduction in lead emissions and may also offset part of the increase in GWP; however, the environmental consequences of the alternative use of lead in this case appear to be less significant than the changes within the solder life cycle. In terms of GWP, the most important consequences are an increase in tin production and in the generation of electricity for solder application.

Attributional and consequential LCAs provide complementary knowledge about the consequences of the shift in solder paste composition. The ALCA describes how the solder life cycle changes. For example, the ALCA results indicate that the Pb-free life cycle contributes 10% more to the GWP than the SnPb life cycle. The CLCA results indicate the possible or foreseeable environmental consequences in a broader context.

It appears difficult to find price elasticity estimates in the literature for the supply and demand of specific metals. Such estimates seem to be available for only a few goods. It may also be problematic to identify the marginal production and use of specific goods, more so for global electricity than for lead, as there are many more potential electricity users than lead users. On a general level, CLCA has been hampered by the lack of easily available marginal data and input data for economic partial equilibrium models. In this context it is important to note that a CLCA model can generate new knowledge, even when important input data are quantitative assumptions, based on a semi-qualitative discussion.

Virtually all intermediate products in a life cycle are traded on a market. The equilibirum between supply and demand on this market must be taken into account in order to accurately describe the consequences of using a specific product, material, chemical, or energy carrier in the life cycle investigated). However, the cumbersome collection or generation of economic data means that partial equilibrium models can be developed for only a few markets. The decision regarding what markets to model in a CLCI can be based on accumulated experience. In this case, we had little relevant experience when the project started. Instead, our decision to focus on the lead and scrap lead markets was based on the fact that lead flow was the political issue at hand. We also used preliminary GWP_{100} results from the attributional study to identify electricity generation as an important aspect of the solder systems, although, in the end, we still were unable to model the electricity market. Using attributional results as a basis on which to identify the aspects requiring investigation in a consequential study is often a practical solution, although it is not without risk. Flows that are important in an ALCI may be less important in a CLCI, or vice versa.

In the practical CLCI modelling, we first attempted to model the consequences of a shift in the contents of solder in a single process tree, as illustrated in Fig. 2. We soon realised that this solution required many manual calculations. Typical LCA software requires that the input data in each unit process be normalised to a reference flow to or from the unit process. The reference flow is generally the product produced in the unit process. When the process tree describes the unit processes associated with a specific product or service, this requirement is not a problem: if the input data are available in terms of annual emissions, etc., they can simply be divided by the annual quantity of the product or similar, to obtain the normalised input data. But when the process tree describes the difference in flows and unit processes between different ways of fulfilling a function, the relation between the emissions and products in each unit process is much more complex. To minimise manual calculations and the associated risk of error, we developed a separate CLCI model for each solder. For the Pb-free solder this model was identical to the ALCI model and, for SnPb solder, the CLCI model was an expansion of the ALCI model.

The environmental consequences of alternative lead use are significant in our model, although less significant than the environmental consequences within the solder life cycle. This can be explained by the fact that a large quantity of lead is required to produce each MWh of electricity from the photovoltaic-plus-battery system. In general, the alternative use of a material is the application that is affected by a change in the availability and price of the material. It is reasonable

to assume that an application where the material cost makes up a large part of the total cost is more sensitive to price changes. This implies that the alternative use of a material is often an application that requires a large quantity of the material for each unit of functional or economic output. It is possible that the alternative use of a kg of a certain material does not replace a great deal and that the environmental consequences of the alternative use are often less significant than the environmental consequences of using the material within the life cycle of the original product. Testing this hypothesis requires experience from much more CLCIs than this study.

Acknowledgement. Financial support for this work has been provided by the EPROPER research school and the Swedish Foundation for Strategic Environmental Research (MISTRA). The authors are also grateful to our data suppliers for making this study possible.

References

[1] Verhoef EV, Reuter MA, Scholte A (2003): A Dynamic LCA Model For Assessing The Impact of Lead Free Solder. In Proceedings of Metallurgical and Materials Processing Principles and Technologies (Yazawa International Symposium) Vol. 2: High-Temperature Metal Production, 2–6 March, San Diego, California, USA, 606

[2] European Union (2003): Directive 2002/95/EC of the European Parliament and of the Council of 27 January 2003 on the restriction of the use of certain hazardous substances in electrical and electronic equipment, Official Journal of The European Union, 13.2.2003: L37/19–23

[3] Tukker A, Buijst H, van Oers L, van der Voet E (2001): Risk to Health and the Environment Related to the Use of Lead in Products. TNO Report STB-01-39, 65

[4] Griese H, Müller J, Reichl H, Somi G, Stevels A, Zuber KH (2000): Environmental Assessment of Lead Free Interconnection Systems. In Proceedings of the Symposium on Lead Free Interconnect Technology, SMTA, 13–14 June, Boston, Massachusetts, USA

[5] Deubzer O, Hamano H, Suga T, Griese H (2001): Lead-free Soldering – Toxicity, energy and resource consumption. In Proceedings of the IEEE International Symposium on Electronics and the Environment, 7–9 May, Denver, Colorado, USA, 290–295

[6] Deubzer O, Griese H, Suga T (2001): Lead-free Soldering – Future aspects of toxicity, energy and resource consumption. In Proceedings of the Second International Symposium on Environmentally Conscious Design and Inverse Manufacturing, 11–15 Dec., Tokyo, Japan, 952–957

[7] Deubzer O, Meysel F, Munoz JR (2003): Modelling the Solder and Resource Consumption in Lead-Free Soldering. In Proceedings of the Third International Symposium on Environmentally Conscious Design and Inverse Manufacturing, 8–11 Dec., Tokyo, Japan, 714–720

[8] Deubzer O, Reichl H, Madsen J, Griese H, van der Wel H (2004): Sustainable Use and Restrictions of Metals in the Electronics Industry. In Proceedings of the Electronics Goes Green, 6–8 Sept., Berlin, Germany, 433–438

[9] Hamano H, Suga T, Okamoto M, Deubzer O (2001): Environmental Impact Evaluation for the full life cycle of products using Pb-free solders. In Proceedings of the Second International Symposium on Environmentally Conscious Design and Inverse Manufacturing, 11–15 Dec., Tokyo, Japan, 1079–1083

[10] Fukuda Y, Pecht MG, Fukuda K, Fukuda S (2003): Lead-Free Soldering in the Japanese Electronics Industry. IEEE Transactions on Components, Packaging and Manufacturing Technology, 26, 616–624

[11] Mihaescu R, Carstea H (2003): Ecological Technologies for Soldering Without Lead in Electronic and Telecommunications – Today and Tomorrow. In Proceedings of 26th International Spring Seminar on Electronics Technology: Integrated Management of Electronic Materials Production, 8–11 May, High Tatras, Slovak Republic, 232–234

[12] Ascencio CA, Madsen JN (2004): Life Cycle Assessment of Lead Free Solders. In Proceedings of the Electronics Goes Green, 6–8 Sept., Berlin, Germany, 427–432

[13] Turbini LJ, Munie GC, Bernier D, Gamalski J, Bergman DW (2001): Examining the environmental impact of lead-free soldering alternatives. IEEE Transactions on Electronics Packaging Manufacturing 24, 4–9

[14] Graedel TE (2002): Material substitution: a resource supply perspective. Resources, Conservation and Recycling 34, 107–115

[15] Verhoef EV, Reuter MA, Scholte A (2003): A Dynamic LCA Model For Assessing The Impact Of Lead Free Solder. In Proceedings of Metallurgical and Materials Processing Principles and Technologies (Yazawa International Symposium), Vol. 2: High-Temperature Metal Production, 2–6 March, San Diego, California, USA, 605–623

[16] Warburg N, Herrmann C, Eyerer P (2003): Lead Free from a Different Stakeholders Point of View. Presentation at IPC SMEMA Council's APEX® 2003, 31 March–2 April, Anaheim, California, USA

[17] Geibig J, Socolof M (2003): Summary of Activities for a Life-Cycle Environmental Impact Evaluation of Tin-Lead and Lead Free Solder. Presentation at IPC SMEMA Council's APEX® 2003, 31 March–2 April, Anaheim, California, USA

[18] Geibig J, Socolof ML (2004): Life-Cycle Impacts of Lead and Lead-Free Solder use in Reflow Soldering of Electronics Manufacturing. In Proceedings of the Electronics Goes Green, 6–8 Sept., Berlin, Germany, 191–196

[19] Ku A, Ogunseitan O, Saphores J, Shapiro A, Schoenung JM (2003): Lead-Free Solders: Issues of Toxicity, Availability and Impacts of Extraction. In Proceedings of the 53rd Electronic Components and Technology Conference, New Orleans, Louisiana, USA, 27–30 May, 47–53

[20] Ogunseitan D, Schoenung J, Saphores J, Shapiro A, Bhuie A, Kang H, Nixon H, Stein A (2003): The devil that we know – Lead (Pb) replacement policies under conditions of scientific uncertainty. The Third International Symposium of Microelectronics and Packaging, June, Israel, 76–87

[21] Ekvall T (1999): System Expansion and Allocation in Life Cycle Assessment – With Implications for Wastepaper Management. PhD thesis, Chalmers University of Technology, Gothenburg, Sweden

[22] Tillman AM (2000): Significance of Decision-Making for LCA Methodology. Environ Impact Assess Rev 20, 113–123

[23] Ekvall T, Weidema BP (2004): System Boundaries and Input Data in Consequential Life Cycle Inventory Analysis. Int J LCA 9, 161–171

[24] Ekvall T, Ciroth A, Hofstetter P, Norris G (2004): Comparative assessment of attributional and consequential methods for LCA, paper presented at 14th Annual Meeting of SETAC-Europe, Prague, Czech Republic, April 2004

[25] Heintz B, Baisnée PF (1992): System Boundaries. In: Life Cycle Assessment – Workshop Report, Leiden, The Netherlands, 35–52

[26] Weidema BP (1993): Development of a Method for Product Life Cycle Assessment with Special References to Food Products (Summary). PhD thesis, Technical University of Denmark, Lyngby, Denmark

[27] Baumann H (1996): LCA Use in Swedish Industry. Int J LCA 1, 122–126

[28] Frischknecht R (1997): Goal and Scope Definition and Inventory Analysis. In: Udo de Haes H, Wrisberg N (eds), Life Cycle Assessment: State-of-the-Art and Research Priorities. LCA Documents, Vol. 1, ecomed publishers, Landsberg, Germany, 59–88

[29] Heijungs R (1997): Economic Drama and the Environmental Stage – Formal Derivation of Algorithmic Tools for Environmental Analysis and Decision Support from a Unified Epistemological Principle. PhD thesis, Leiden University, The Netherlands

[30] Baumann H (1998): Life Cycle Assessment and Decision Making – Theories and Practices. PhD thesis, Chalmers University of Technology, Gothenburg, Sweden

[31] Cowell SJ (1998): Environmental Life Cycle Assessment of Agricultural Systems: Integration Into Decision-Making. PhD thesis. University of Surrey, Guildford, UK

[32] Hofstetter P (1998): Perspectives in Life Cycle Impact Assessment – A Structured Approach to Combine Models of the Technosphere, Ecosphere and Valuesphere. Kluwer Academic Publishers, Dordrecht

[33] Curran MA, Mann M, Norris G (2005): The International Workshop on Electricity Data for Life Cycle Inventories. J Cleaner Prod 13, 853–862

[34] Industrial Research and Development Corporation (2004): Personal communication with Jerry Börjesson, engineer, Mölndal, Sweden, 2004-05-17

[35] Frischknecht R, Hofstetter P, Knoepel I, Meénard M, Dones R, Zollinger E (1994): Ökoinventare für Energiesysteme. Zürich: Bundesamt für Energiewirtschaft. Data adapted by CIT Ekologik to LCI data for cradle-to-gate hard coal production

[36] Frischknecht R, Hofstetter P, Knoepel I, Meénard M, Dones R, Zollinger E (1994): Ökoinventare für Energiesysteme. Zürich: Bundesamt für Energiewirtschaft. Data adapted by CIT Ekologik to LCI data for cradle-to-gate heavy fuel oil production

[37] Frischknecht R, Hofstetter P, Knoepel I, Meénard M, Dones R, Zollinger E (1994): Ökoinventare für Energiesysteme. Zürich: Bundesamt für Energiewirtschaft. Data adapted by CIT Ekologik to LCI data for cradle-to-gate diesel production

[38] International Energy Agency (2001): Key World Energy Statistics 2003. 2001 Fuel Shares of Electricity Generation, 24

[39] Vattenfall AB (2000): Environmental product declaration for 'Electricity from the River Lule Älv', URL: <http://www.environdec.com. Accessed 2004-12-07>

[40] Vattenfall AB (1998): Environmental product declaration for 'Electricity from the Nuclear Power Plant at Forsmark'. URL: <http://www.environdec.com>. Accessed 2004-12-07

[41] Frischknecht R (1996): Ökoinventare für Energiesysteme. Zürich, Bundesamt für Energiewirtschaft. Data adapted by CIT Ekologik to LCI data for gate-to-gate hard coal combustion

[42] Bakkane KK (1994): Life cycle data for Norwegian oil and gas. S.l.: Tapir/Norwegian Inst. of Technology. Data adapted by CIT Ekologik to LCI data for cradle-to-gate natural gas production

[43] Frischknecht R (1996): Ökoinventare für Energiesysteme. Zürich: Bundesamt für Energiewirtschaft. Data adapted by CIT Ekologik to LCI data for gate-to-gate natural gas combustion

[44] Frischknecht R (1996): Ökoinventare für Energiesysteme. Zürich, Bundesamt für Energiewirtschaft. Data adapted by CIT Ekologik to LCI data for gate-to-gate heavy fuel oil combustion

[45] Ecoinvent database (2004): Tin, at regional storage, Index number 1155, URL: <http://www.ecoinvent.org, Accessed 2004-02-20>

[46] Boustead I, Panvalkar SG (1998): Ecoprofile of Primary Copper Production. A Report for the International Copper Association, January 1998. Boustead Consulting Ltd. Recovery of silver from copper production, 71–73

[47] Ecoinvent database (2004): Lead, at regional storage, Index number 1103, URL: <http://www.ecoinvent.org, Accessed 2004-02-20>

[48] Ecoinvent database (2004): Copper, primary, couple production nickel, GLO, Index number 1089, URL: <http://www.ecoinvent.org, Accessed 2004-02-20>

[49] Confidential source (2000): Battery final assembly gate-to-gate in Switzerland

[50] Alsema E (2000): Environmental life cycle assessment of solar home systems. Report NWS-E-2000-15, Dept. of Science Technology & Society, Utrecht University, 23–24, 75–76

[51] Berg S (1996): Emissioner till luft från fossila bränslen i svenskt skogsbruk, En inventering för LCA av träprodukter. (In Swedish) 7. Data adapted by CIT Ekologik to LCI data for gate-to-gate diesel combustion

[52] Alsema E (2000): Environmental life cycle assessment of solar home systems. Report NWS-E-2000-15, Dept. of Science Technology & Society, Utrecht University, 66

[53] Boliden Bergsöe AB (2002): Environmental report for 2001, Landskrona, Sweden. Pb recycling 11, 14–16

[54] Green Design Initiative (2000): Economic Input-Output Life Cycle Assessment Model, Carnegie Mellon University, Pittsburgh, USA URL: <http://www.eiolca.net> on 'chemicals and chemical preparations, n.e.c.' Accessed 2004-12-07

[55] Mouser Electronics (2004): The price of solder flux, URL: <http://www.mouser.com/catalog/618/1267.pdf>, SP 44 Paste Flux. Accessed 2004-06-28

[56] Höganäs AB (2002): Environmental report for 2001 for Höganäs Halmstadsverken, Halmstad, Sweden. Iron powder production, 7,10,12

[57] Geibig J, Socolof M (2003): Summary of Activities for a Life-Cycle Environmental Impact Evaluation of Tin-Lead and Lead Free Solder. Presentation at IPC SMEMA Council's APEX® 2003, 31 March–2 April, Anaheim, California, USA. Powerpoint presentation, energy inputs for solder paste manufacturing were obtained from slide 13 of 31

[58] Geibig J, Socolof M (2003): Summary of Activities for a Life-Cycle Environmental Impact Evaluation of Tin-Lead and Lead Free Solder. Presentation at IPC SMEMA Council's APEX® 2003, 31 March–2 April, Anaheim, California, USA. Powerpoint presentation. Reflow oven power consumption was obtained from slide 18 of 31

[59] Novastar Inc. (2004): Solder paste application regarding length of reflow soldering oven. URL: <http://www.novastarinc.com/specs/2000spec.pdf>. Accessed 2004-12-07

[60] Ascom Tateco AB (2004): Personal communication with Bert-Ove Johansson, Herrljunga, Sweden, regarding maximum number of boards in reflow solder oven, 2004-03-04

[61] Chalmers University of Technology (2004): Personal communication with Ph.D. student Cristina Andersson regarding soldering process time, 2005-05-18

[62] Ragn-Sells Elektronikåtervinning AB (2004): Personal communication with Per Frisell at Vällingby, Sweden, regarding dismantling electricity consumption per mass of electronics, 2004-05-18

[63] Renova AB (2003): Environmental report for 2002 from combined heat and power plant driven by Renova AB at Sävenäs, Sweden, 4–54

[64] Vafab (2003): Environmental report for 2002 from waste treatment facility Sänkmossen 1982-60-001, Fagersta, Sweden. URL: <http://www.vafab.se/foretag/Smnmilr03.htm>. Accessed 2004-12-07

[65] Henkel Technologies (2003): Personal communication with Dr. Hector Steen, United Kingdom, 2003-12-05

[66] Organisation for Economic Co-operation and Development (1995): Recycling of copper, lead and zinc bearing wastes. OECD, Paris

[67] Brady TA, McManus TJ (2004): A DfE?? Framework for Sustainable IT Products. In Proceedings of the Electronics Goes Green, 6-8 Sept., Berlin, Germany, 197–202

[68] Weidema BP, Frees N, Nielsen AM (1999): Marginal Production Technologies for Life Cycle Inventories. Int J LCA 4, 48–56

[69] Lead Development Association International (2004): URL: <http://www.ldaint.org/technotes.htm>. Accessed 2004-09-03

[70] Organisation for Economic Co-operation and Development (1993): Lead – Background and National Experience with reducing risk, No. 1, Env. Mono. No. 67, OCDE/GD (93)67, 47. URL: <http://www.oecd.org/document/22/0,2340,en_2649_34375_1935254_1_1_1_1,00.html>. Accessed 2004-09-23

[71] Thornton I, Rautiu R, Brush S (2001): Lead – the facts. Lead Development Association International, London, UK. URL: <http://www.ldaint.org/factbook/>. Accessed 2004-08-18

[72] Global sources (2004): Lead drives SLA battery price hikes. URL: <http://www.globalsources.com/am/article_id/9000000049832/page/showarticle?action=GetArticle/>. Accessed 2004-08-27

[73] Frost & Sullivan Growth Consulting (2004): URL: <http://www.frost.com/prod/servlet/press-release.pag?docid=22202816>. Accessed 2004-12-14

[74] Jackson J (2004): General considerations in metal and fuel markets. URL: <http://web.pdx.edu/~jjackson/MetalsFuelsMarkets.html>. Accessed 2004-12-14

[75] Sveriges Geologiska Undersökning (2004): Sammanställning av vissa valutakurser, räntor, ädelmetall-, basmetall- och oljepriser, June 2004, Geological Survey of Sweden, Uppsala, Sweden (in Swedish)

[76] Backman CM (1997): En analys av tennbranschen och pris-variationer på tenn utgående från M.E. Porters modell av branschkonkurrensen. Uppsala University, Uppsala, Sweden

[77] Organisation for Economic Co-operation and Development (1993): Lead – Background and National Experience with reducing risk. No. 1, Env. Mono. No. 67, OCDE/GD (93) 67, 65, URL: <http://www.oecd.org/document/22/0,2340,en_2649_34375_1935254_1_1_1_1,00.html>. Accessed 2004-09-23

[78] Vest H (2002): Fundamentals of the Recycling of Lead-Acid Batteries. Gate Information Service, 1–2. URL: <http://www.gtz.de/energie/download/Recycling_of_batteries.pdf>. Accessed 2004-09-22

[79] Organisation for Economic Co-operation and Development (1993): Lead – Background and National Experience with reducing risk. No. 1, Env. Mono. No. 67, OCDE/GD (93) 67, 65–66, URL: <http://www.oecd.org/document/22/0,2340,en_2649_34375_1935254_1_1_1_1,00.html>. Accessed 2004-09-23

[80] Intergovernmental Panel on Climate Change (2001): Climate Change 2001: The Scientific Basis. Cambridge, UK, Cambridge University Press, Technical Summary of the Working Group I Report, 47

[81] Swedish Environmental Protection Agency (1992): Characterisation factors for GWP

[82] The Silver Institute (2004): URL: <http://www.silverinstitute.org/supply/production.php>. Accessed 2004-12-07

Received: January 28th, 2005
Accepted: May 23rd, 2005
OnlineFirst: May 23rd, 2005